Management and Conservation Article

Fuel-Reduction Treatment Effects on Avian Community Structure and Diversity

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ABSTRACT We assessed responses of the breeding bird community to mechanical thinning and prescribed surface fire, alone and in combination, between 2000 and 2006 in ponderosa pine (*Pinus ponderosa*) forests in northern Arizona, USA. Fuel-reduction treatments did not affect species richness or evenness, and effects on density of 5 commonly detected species varied among species. Populations of some species, such as the western bluebird (*Sialia mexicana*), increased following burning treatments, whereas others, such as the mountain chickadee (*Poecile gambeli*), decreased in response to thinning treatments. Our results also identified a temporal response component, where avian community composition and structure changed synchronously on all treatments over time. Given the modest effects these small-scale fuel-reduction treatments had on avian composition and the specific density responses of particular species, our results suggest that land managers should consider implementing prescribed surface fire after thinning projects, where appropriate. (JOURNAL OF WILDLIFE MANAGEMENT 72(5):1168–1174; 2008)

Many southwestern ponderosa pine (*Pinus ponderosa*) forests in the United States were characterized by park-like stands prior to Euro–American settlement in the late 1800s (Harrington and Sackett 1990, Stone et al. 1999). This open structure was maintained by low-intensity surface fires that burned at intervals of 2–20 years prior to a management paradigm focused on fire suppression (Harrington and Sackett 1990, Covington and Moore 1994). Fire, along with other natural episodic events such as drought, insect infestation, and pine regeneration created a natural range of variability across a heterogeneous landscape (Allen et al. 2002). Interruption of the natural disturbance regime coupled with human activities, such as logging and grazing, caused substantial changes to the composition and structure of southwestern ponderosa pine forests (Covington et al. 1997, Allen et al. 2002, Youngblood et al. 2004). For example, exclusion of fire allowed accumulation of surface and ladder fuels, and intensive logging removed many of the larger diameter trees, leaving behind relatively homogeneous stands of small-diameter trees (Covington and Moore 1994, Fulé et al. 1997, Allen et al. 2002, Agee and Skinner 2005). Grazing practices resulted in reductions in the herbaceous cover and fine fuels that facilitated the spread of naturally caused low-intensity surface fires (Harrington and Sackett 1990, Allen et al. 2002). These factors, combined with several exceptionally wet years in the early 1900s, have led to forest stands with dense forest structures well outside the range of natural variability (Cooper 1960, Covington and Moore 1994, Agee and Skinner 2005).

In the past decade, the severity and size of wildfires in the Southwest have increased dramatically, leading to changes in forest management that focus on restoration including implementation of fuel-reduction treatments such as thinning and burning across large areas (Covington 2000, Allen et al. 2002, Agee and Skinner 2005, Noss et al. 2006). Although scientists are still gaining information on effects of mechanical thinning and prescribed fire on wildlife, these treatments are being used increasingly in the Southwest to reduce fuel accumulations and fire risk. In some cases, mechanical thinning and prescribed fire have been implemented so as to mimic natural disturbance and to recreate forest structural characteristics that existed prior to Euro–American settlement (Covington 2000, Meyer et al. 2001). Often, such restoration treatments are assumed to provide favorable habitat conditions for native biota because they return ecosystem processes to some reference or natural condition (Block et al. 2001). However, these assumptions remain largely untested (Block et al. 2001).

Several studies in the Southwest have evaluated effects of prescribed fire, thinning treatments of various intensities, or a combination of thinning and burning in a restoration framework (Szaro and Balda 1986, Horton and Mannan 1988, King and DeGraaf 2000, Germaine and Germaine 2002). Much of our understanding of effects of fire on bird communities, however, is based on observational studies after postfire salvage logging or wildfire (Kotliar et al. 2002, Saab et al. 2004, Bock and Block 2005, Saab and Powell 2005). We reviewed the literature to frame expectations about predicted responses of 5 common species to treatments. Focal species included dark-eyed junco (*Junco hyemalis*), mountain chickadee (*Poecile gambeli*), pygmy

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nuthatch (*Sitta pygmaea*), western bluebird (*Sialia mexicana*), and yellow-rumped warbler (*Dendroica coronata*). We hypothesized that western bluebird densities would increase in burned areas because previous research showed an increase in abundance after fire (Overturf 1979, Blake 1982, Szaro and Balda 1986). As insectivores, western bluebirds may benefit from increased insect populations following fire (Guinan et al. 2000, Saint-Germain et al. 2004). We expected a negative response by mountain chickadees because they are typically more abundant in unburned areas and have been shown to respond negatively to severe wildfire (Aulenbach and O'Shea-Stone 1983, Johnson and Wauer 1996, Saab and Powell 2005). For dark-eyed junco, pygmy nuthatch, and yellow-rumped warbler, we expected a mixed response to treatments because previous research showed equivocal responses to wildfire and fuels treatments (Lowe et al. 1978, Bock and Bock 1983, Saab and Powell 2005).

To better understand how fuel-reduction treatments affect passerines in ponderosa pine forests, our objectives were to quantify changes in avian diversity responses to forest fuel-reduction treatments and to more closely evaluate the population-level response of several common species. Specifically, we examined whether: 1) fuel-reduction treatments affect avian diversity, 2) avian community composition and structure change over time, and 3) fuel-reduction treatments affect density of more common species. Accordingly, we developed a series of predictions for selected species, but we anticipated considerable variation in density response among treatments and species.

STUDY AREA

We conducted our study at 3 sites in northern Arizona, USA, located on the Kaibab (KA Hill: 35°12.0′33.9″ latitude, 111°44.0′32.2″ longitude) and Coconino (Powerline: 35°12.0′33.9″ latitude, 111°45.0′32.2″ longitude; Rudd's Tank: 35°14.0′05.9″ latitude, 111°44.0′58.4″ longitude) National Forests. Overstory at these sites was dominated by ponderosa pine, but also included Gambel oak (*Quercus gambelii*), one-seed juniper (*Juniperus monosperma*), and alligator juniper (*J. deppeana*). Mature, yellow-bark ponderosa pine trees were present in small numbers, whereas small-diameter (dbh <25 cm) trees were common at all sites resulting from previous timber harvest activities (Dickson et al. 2004). Arizona fescue (*Festuca arizonica*) and blue grama (*Bouteloua gracilis*) were the most common plants in the understory vegetation. Topography at these sites was mostly flat with mean elevation of 2,193 m (±100 SD). Average annual temperature was 6.5° C (±10.6 SD) and average annual precipitation was 53.1 cm (±0.8) for the period 2005–2006 (Huebner 2006).

METHODS

Experimental Design

To assess the response of the avian community to fuel-reduction treatments, we used a modified Before-After–Control-Impact experimental design (Green 1979; Stewart-
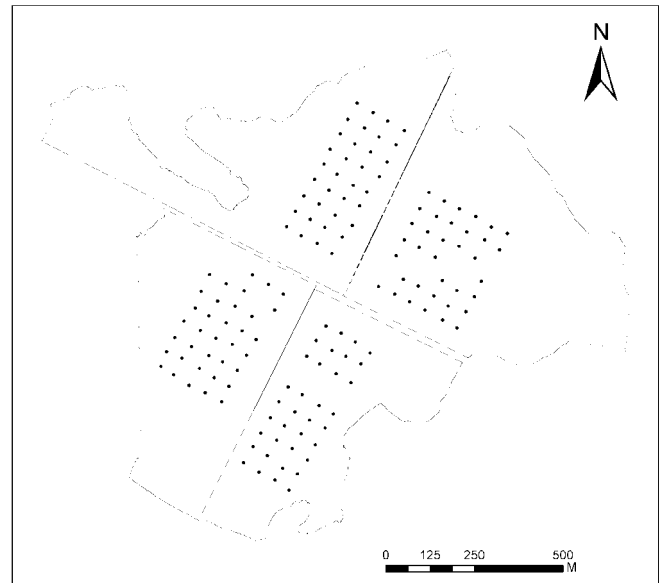

Figure 1. Map illustrating the juxtaposition of treatments within a block and the 36-point sampling array in each treatment unit for one of the 3 replicate blocks in northern Arizona, USA, 2000–2006.

Oaten et al. 1986, 1992). Each of the 3 study sites was established as a block ($n = 3$ replicates) within which each of 4 experimental fuel-reduction treatments was applied: a control, thin only, prescribed fire only, and thin followed by prescribed fire (Fig. 1). Treatments were assigned to units within the block at random; however in 2 blocks random treatment locations were constrained to 1 of 2 (rather than 1 of 4) units to reduce cost of treatment implementation (J. Bailey, Oregon State University, personal communication). Treatments were chosen to represent currently implemented treatments in many southwestern ponderosa pine forests. Treatment units ranged in size from 16 ha to 30 ha. United States Forest Service district personnel implemented thinning treatments in autumn 2002 and prescribed surface fire treatments in autumn 2003 for each site using strip-head fire techniques (Faiella and Bailey 2007). Treatments aimed to reduce stem density to 116 trees/ha and basal area to 13 m²/ha (Faiella and Bailey 2007).

Avian Sampling

To evaluate treatment effects on avian composition, diversity, and density, we surveyed birds during the breeding season (late May through early Jul) of each sampling year. We surveyed birds between 2000 and 2002 for pretreatment data and between 2003 and 2006 for posttreatment data. We established a 36-point sampling array within each treatment unit, typically as a 6 × 6 grid, with 50 m between points, which comprised a 10-ha core sampling area (Fig. 1). Regardless of array shape, we maintained a buffer zone of ≥50 m between points and the edge of the treatment.

To quantify how forest treatments influenced avian community structure, we used estimates of species richness based on first-order jackknife estimator and evenness based on Simpson's index (inverse, $1/D$), a measure that increases as community assemblage becomes more even (Magurran

Table 1. Pretreatment (2000–2001) and post-treatment (2004–2005) average basal area and stem density (trees/ha) measured across 3 sites in northern Arizona, USA. We averaged values by treatment (control, thin only, burn only, thin and burn).

| Treatment type | Basal area (m²/ha) | | | | Trees/ha | | | |
| | Pretreatment | | Post-treatment | | Pretreatment | | Post-treatment | |
	$\bar{x}$	SD	$\bar{x}$	SD	$\bar{x}$	SD	$\bar{x}$	SD
Control	30.3	4.2	31.2	4.8	674	258	664	263
Thin only	29.9	2.2	14.8	1.6	592	213	194	41
Burn only	30.1	3.7	30.4	2.9	618	295	604	282
Thin and burn	24.9	4.4	13.0	3.4	451	151	151	28

2004). We calculated diversity measures for each treatment unit for all sites and all years sampled using the EstimateS Software Program (V7.5; Colwell 2005). We estimated mean and variance for species richness and species evenness using 1,000 bootstrap randomizations (Sokal and Rohlf 1995). We analyzed species richness estimates and Simpson's diversity indices for treatment effects in an analysis of variance that included site as a random blocking factor and treatment, treatment period, and years within treatment period as fixed factors. We considered estimates for multiple years on the same treatment unit within site repeated measurements. We used interactions between treatment and treatment period to gauge treatment effects.

We used nonmetric multidimensional scaling (NMDS) with Bray–Curtis distances to examine avian species composition and abundance by treatment (McCune and Mefford 1999). Pretreatment data were pooled among years and each additional year represented a year since treatment implementation. We then used Bray–Curtis dissimilarity to assess differences in avian community composition among treatments, where parameter estimates indicate the relative importance of corresponding environmental variables (Dyer 1978). Environmental variables included in this analysis were treatment, site, and year. Each data point in the NMDS plot represented the relative abundance of all species present in a given treatment unit for a given year.

We used a distance-based sampling approach to estimate avian density before and after treatment (Buckland et al. 2001). At each sampling point, we recorded all bird species detected by sight or sound during a 5-minute period but excluded from analysis birds that flew across the sample area. We estimated distance from the observer to each bird up to ≤50 m. We conducted surveys between 30 minutes after sunrise and 1000 hours, with 9–12 points visited per day. Using this sampling scheme, we made 3–4 visits to each treatment unit spaced over the breeding season, with each point visited only once. We arranged sampling points into transects to minimize sampling radius overlap on any given day, and we selected transects at random. Use of distance sampling allowed us to relax assumptions of spatial independence among our closely spaced sampling points because distance-based models are robust to detection of the same individuals during different sampling periods or at >1 sampling station (Buckland et al. 2001).

For each focal species, we used Program DISTANCE (V5.0; Thomas et al. 2005) to estimate density (individuals/

10 ha) by treatment. Because thin and burn treatments were completed in different years, we computed densities separately for 3 periods: a pretreatment period (2000–2002), a post-thin period (2003–2006), and a post-thin and post-burn period (2004–2006). For each species, we considered year, study site, and treatment type as covariate factors in the estimation of a global detection probability using the multiple covariate distance sampling (MCDS) analysis engine in DISTANCE (see Buckland et al. 2004, Marques and Buckland 2004). We used MCDS instead of conventional distance sampling analysis because we required species- and stratum-specific estimates, and because the MCDS approach was more robust to small sample size (Marques and Buckland 2004). Importantly, because we used a global detection function to estimate species density by treatment, our stratified estimates are not independent and have associated variance estimates that are biased low (Buckland et al. 2001). We used Akaike's Information Criterion (AIC) to select a model for density from a candidate set that included all possible factor combinations (Burnham and Anderson 2002). When differences in AIC_c values for competing models were ≤2, we averaged density estimates and computed unconditional standard errors (Burnham and Anderson 2002).

To quantify the magnitude of change in estimated density of each focal species to individual treatments in the pre- and post-treatment periods, we estimated the difference in density (Δ-pre and Δ-post, respectively) by subtracting control density from treatment density. We then calculated a standardized difference (Δ*) by subtracting Δ-pretreatment density from Δ-post-treatment to determine the cumulative response (or the difference in differences) to a treatment by a given species. We considered this measure equivalent to the time × treatment interaction.

RESULTS

Although estimates varied among replicates, basal area of control and burn-only treatments were virtually unchanged after treatment (Table 1). Basal area of thin-only and thin-and-burn treatments was reduced by approximately 50% (Faiella and Bailey 2007; J. Bailey, unpublished data). Tree density was unchanged after treatment in control and reduced <10% in burn-only treatments (Faiella and Bailey 2007). However, tree density was reduced by approximately 60–70% in thin-only and thin-and-burn treatments (Faiella and Bailey 2007; J. Bailey, unpublished data).
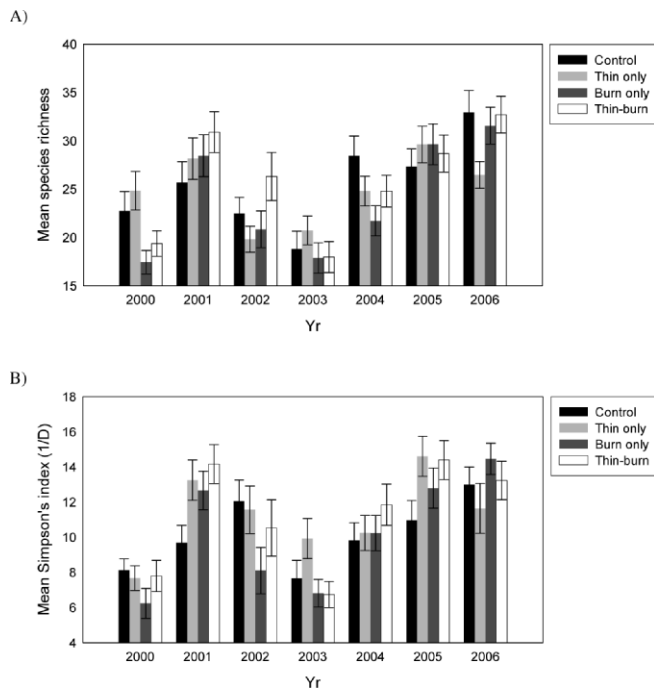
A)

B)

Figure 2. Stratum-level mean (±SE) estimates of (A) species richness and (B) Simpson's index (inverse) for species evenness for 63 avian species over 7,363 detections sampled at the Southwest Plateau sites in northern Arizona, USA, May–July, 2000–2006. We generated estimates using 1,000 bootstrap randomizations.

We recorded 7,363 detections among 63 avian species during the 2000–2006 breeding seasons. Species richness ($F_{3,8} = 0.15$; $P = 0.93$) and evenness ($F_{3,8} = 0.52$; $P = 0.68$) did not change in response to treatment (Fig. 2).

Although there was little difference in the avian communities among treatments, communities in all 4 treatments followed the same pattern of movement through ordination space over time (Fig. 3). This synchronous change in the avian community through time was reflected in the amount of dissimilarity among treatments, which averaged 0.41 ± 0.003 (SE) and increased to 0.51 ± 0.004 between years. Differences in avian community structure were 6 times greater among years than among sites, and 8 times greater than among treatments.

Responses of focal species to treatments varied considerably. Changes in dark-eyed junco density ranged from a 9% decrease to a 29% increase on the thin-only and burn-only treatments, respectively, relative to control (Fig. 4). In contrast, mountain chickadee densities declined on all treatments, most on the thin-only treatment, where density was reduced by >50% during the posttreatment period to 2.3 individuals/10 ha (±0.8 SE). Pygmy nuthatch densities remained constant across treatments except the thin-and-burn treatments, where densities increased by >500%. Western bluebird densities increased on all treatments with densities more than doubling on the burn-only and thin-and-burn treatments. Yellow-rumped warbler demonstrated a mixed response to treatments, with the largest change in density on the thin-and-burn treatment, which decreased by 100%.
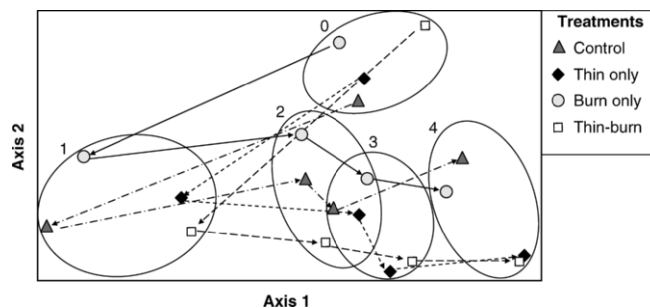

Figure 3. Ordination of pre- and post-treatment avian community data using nonmetric multidimensional scaling by treatment, northern Arizona, USA, 2000–2006. We pooled pretreatment avian community data by treatment for all years. We pooled post-treatment data by treatment for each sampling year. Numbers indicate sampling year (i.e., 0 = pretreatment, 2000–2002; 1 = first yr post-treatment, 2003; 2 = second yr post-treatment, 2004, etc.).

DISCUSSION

Treatments to reduce forest fuels across the range we studied had little effect on avian diversity over our 4-year study but did affect some aspects of species composition and abundance. High annual variation, however, obscured clear and consistent patterns. Our results suggest that although the small-scale forest treatments we studied may have influenced the avian species present, natural annual variation in density was a stronger source of variation. In contrast to our results, Szaro and Balda (1986) found that both weather and various intensities of forest-thinning treatments influenced bird density and species richness but that treatments had a greater influence on community composition.

Fuel-reduction treatments in ponderosa pine forests affected the focal species in different ways. Increases in western bluebird densities on treatments that included fire (i.e., burn-only and thin-and-burn treatments) likely resulted from a combination of opening the understory vegetation and an increase in foraging opportunities due to burning. Western bluebird is commonly associated with sites that have recently burned, and our results indicate a strong positive response to both thinning and burning treatments (Overturf 1979, Bock and Block 2005, Saab and Powell 2005). Wightman and Germaine (2006) suggest that restoration treatments that decrease tree densities and increase herbaceous vegetation may provide a more abundant food resource and generally improve habitat quality for bluebirds, an explanation consistent with our findings.

Given the low number of detections for mountain chickadee, it was difficult to calculate density of this species as it decreased on all treatments and appeared most sensitive to the thin-only treatment. Mountain chickadee has been shown to be negatively affected by different timber management practices, and the reductions in density we observed are consistent with previous research in Arizona mixed-conifer forests (Franzreb 1978, McCallum et al. 1999).

The magnitude of density change by pygmy nuthatches on thin-and-burn treatments was unexpected. Several studies
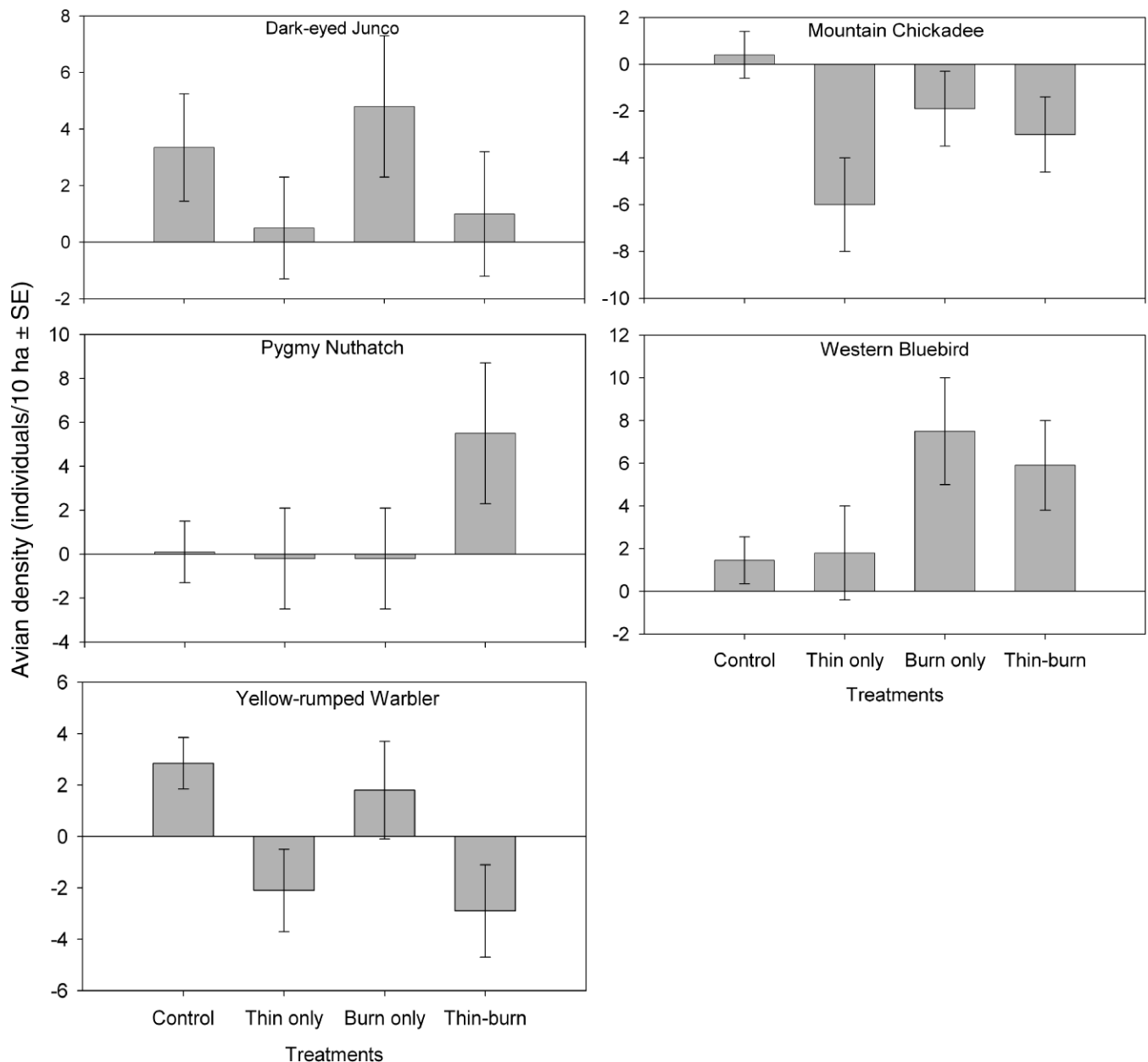
Figure 4. Differences in stratum-level density estimates (individuals/10 ha ± SE) for 5 focal avian species (dark-eyed junco, mountain chickadee, pygmy nuthatch, western bluebird, and yellow-rumped warbler) on the 3 replicate blocks in northern Arizona, USA, 2000–2006. We calculated cumulative response (Δ*) as the difference between post- and pretreatment density for each species by treatment. A positive number indicates an increase and a negative number indicates a decrease in density.

have found that abundance of this species was reduced after wildfires (Lowe et al. 1978, Overturf 1979, Blake 1982, Bock and Block 2005). Removal of snags by wildfire events is usually responsible for the decline seen in pygmy nuthatch densities. However, the high density of snags on the study sites, and the persistence of many snags following treatments, suggests that prescribed fire and thinning, as implemented on these experimental treatments did not have the negative impacts expected following wildfire.

Our fine-scaled, replicated experiment provides a detailed look at short-term responses in density to small-scale changes in forest structure following management treatments common in the Southwest. Results from small-scale plots such as these may also be confounded by the close proximity of treatments, such that some bird species may be responding to the collection of treatments rather than individual treatments. Notably, results of our study are consistent with emerging, large-scale studies of avian community response to forest treatments in the region (e.g., Short 2003, Dickson 2006, Pope 2006). Collectively, this work suggests that as managers move forward with landscape-level forest treatments designed to reduce fire risk and reintroduce appropriate fire as a critical forest process, overall effects on forest passerine assemblages are likely to be modest.

MANAGEMENT IMPLICATIONS

Species that responded positively to thinning treatments used areas that had an average basal area that ranged between 12–15 m^2/ha and stem density between 140–189 trees/ha across sites. We recommend that future small-scale fuel-reduction treatments in southwestern ponderosa pine forests be implemented within this range of forest characteristics, values that are also consistent with the range of 1887 conditions reconstructed for a site in northern Arizona (Fulé et al. 2002). As many managers attempt to reduce fire risk and reintroduce natural processes such as fire back into the ecosystem, our results indicate that burning should be conducted after thinning, because several focal species increased only after completion of both thinning and burning treatments. However, effects on the entire avian community should be considered prior to implementation of forest fuel-reduction programs because the particular species present and their abundances may affect decisions regarding the appropriate treatment to deploy and its location, to better meet management goals. Given the difficulty of managing for many species with variable responses to forest manipulations, creating a mosaic of forest conditions following treatments may be the most suitable approach for a wide range of forest passerines.

ACKNOWLEDGMENTS

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